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High carbon losses from established growing sites delay the carbon sequestration benefits of street tree plantings - a case study in Helsinki, Finland

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Keywords

carbon sequestration

tree soil carbon

tree biomass equations

urban trees

List of abbreviations

AB	aboveground biomass (stem, branches and leaves)
ABW	aboveground woody biomass (stem and branches)
BE	biomass equation (see Appendix)
C	carbon
DBH	diameter at breast height
LOI	loss-on-ignition
O_e	loss-on-ignition estimated (Eq.6) for the periods between LOI samplings
SOM	soil organic matter
SW	soil water
T_f	soil temperature, as measured at tree sites
WB	woody biomass
W_f	soil water content, % of weight, as measured at tree sites
α	LOI and SWC response parameter in the incubation model
β	temperature response parameter in the incubation model

Abstract

We assessed the net carbon (C) sequestration dynamics of street tree plantings based on 10 years of measurements at two case study sites each with different tree species in Helsinki, Finland. We assessed C loss from tree soils and tree C accumulation, tested the applicability of pre-existing growth and biomass equations against observations, and estimated the time point for the beginning of net C sequestration for the studied street tree plantings. The tree woody biomass C accumulation in the first 10 years after planting was 18--32 kg per tree. At the same time the C loss from the growth media was at least 170 kg per growth media volume (25 m^3) per tree. If this soil C loss was accounted for, the net C sequestration would begin, at best, approximately 30 years after planting. Biomass equations developed for traditional forests predicted more stem biomass and less leaf and branch biomass than measured for the species examined, but total aboveground biomass was generally well predicted.

Introduction

Carbon (C) sequestration is one of the ecosystem services that encourage the planting of urban trees (McPherson *et al.*, 2005). The C sequestration effects of urban tree plantings consists of C stock change in above- and belowground tree compartments, and soil organic matter (SOM) stock changes related to tree planting and litter production by planted trees. The belowground C stock of urban trees and soils is not well known, but there are indications that urban soil C stocks can be substantial (Pataki *et al.*, 2006). In the traditional, non-urban forests of Scandinavia, the soil C stock occurs predominantly in the superficial layers and is as large as or larger than that of the vegetation (Liski *et al.*, 2006). In urban greening, trees are planted traditionally in limited container-like soil spaces or wider structural soil (e.g. Grabosky and Bassuk 1995, Neal and Whitlow 1997, Kristoffersen 1999) volumes in which the load-bearing properties of the soil have been enhanced with stony matrices. Fine soil, suitable for tree rooting, is located in the voids of the stone matrix. In both of these methods, artificial growing media brings C-rich soil into the deep layers. Currently, Finnish municipalities use SOM contents of 10--12% (measured as loss-on-ignition, LOI) throughout the standard 1-m-deep growth media in tree plantings (Rakennustietosäätiö, 2010). A square metre of new traditional tree growth media thus typically has a C stock of approximately 40--50 kg C m^{-2} and a structural soil of 10--20 kg C m^{-2}; about 2--10 times more than in traditional upland forest soils in Finland (Liski *et al.*, 2006).

In contrast to natural SOM, which has substantial proportions of slowly decomposing fractions, the artificial growth media organic matter may decompose quickly and lose C to the atmosphere (Bernal *et al.*, 1998). Soil sealing (by e.g. asphalt or pavement), common in urban environments and predominantly used in combination with structural tree soils, impairs soil heat and soil water (SW) exchange (Scalenghe and Marsan, 2009) and limits the C input from above the ground, affecting biomass accumulation and decomposition. These effects may lead to overall C loss from street tree plantings unless the C sequestered

by the tree exceeds the C loss from the growing media. The organic matter in the growing media may be derived from peat, or partially or entirely from renewable C sources, such as compost. In addition to its use in growing media as such, peat is a common additive used when composting sewage sludge, kitchen and food waste etc. (e.g. Himanen and Hänninen 2011), and consequently, also the SOM in compost-derived growing media tends to contain some fraction of peat. While renewable C originating from short-term biogenic cycle is commonly considered neutral in regards to climate change, peat-derived CO_2 in energy production is currently viewed as equivalent to fossil fuel emissions (IPCC 2006). IPCC takes no stand on other uses of peat however; thus the official C accounting status of peat in growing media is somewhat unclear. There is a general interest in finding replacements for peat in the growing media industry however, due to both wetland protection and CO_2 emissions (e.g. Defra 2009).

In a street tree planting, growing media C emissions can be compensated and exceeded by the C sequestration of trees over time. Unfortunately, the size and C stock of urban trees of a given age are not easily predicted (Peper *et al.*, 2014). Currently, aboveground C stocks in urban trees are estimated with allometric tree biomass equations (BEs) developed in traditional forests, if urban-based equations are not available. Root biomass may then be estimated from a set root-shoot ratio despite its large variation between ecosystems and species (e.g. IPCC 2006). However, the accuracy of traditional forest based BEs in an urban context has been questioned (McHale et al., 2009). The particular above- and belowground environments of trees influence both the overall growth rate and biomass distribution within trees (Litton *et al.*, 2007; Zhou *et al.*, 2014). Consequently, the urban environment may lead to biomass distributions different from those observed in traditional forests, with consequent biomass estimation problems. In addition to biomass distribution, the tree-related C inputs into urban soil remain largely unknown. Root exudates and litter likely contribute to soil C stock while, especially in paved areas, the aboveground litter might not, because it is either removed or moves along, unable to enter the soil under the pavement.

The purpose of this study is to estimate the long term carbon dynamics of a street tree planting in the hemi-boreal city of Helsinki. We collected data from two case study street tree plantings (established according to current establishment practices in Finland) about long term C stock changes in the growth media and trees. These were combined with literature based tree growth equations that we tested with separate tree data from different aged street tree plantings in Helsinki, and literature based biomass equations. At the case study sites, we assessed the soil C stock changes occurring during the first 10 years after planting, using a LOI change-based approach. We evaluated the estimate with CO_2 production of soil samples in an incubation experiment. At these sites we estimated the tree biomass accumulation from direct measurements of the case study trees. The measurements were compared against literature-based biomass equations to find the equations corresponding best to the case study observations.

We used the data to estimate the time needed for street plantings to reach the C compensation point (the number of years after planting required for the tree C capture to reach the sum of the soil C loss in the first decade after planting) in Helsinki. Our hypothesis was that the amount of C lost from the growth media of the case study trees would offset the C uptake of the tree growth during our study period, and an improvement in average street tree life expectancy would be needed to obtain C sequestration benefits with current planting practices.

Materials and methods

Case study sites

We studied the tree biomass and soil C changes on two separate street sites, located 800 meters apart from each other, in the Viikki suburb in Helsinki, Finland, (N60˚15', E25°03') over 10 years after the establishment of the street. One street, 250 m in length, was planted with 15 common lime *Tilia* × *vulgaris* Hayne trees (Tilia site) and the other (200 m in length) with 22 black alder *Alnus glutinosa* (L.) Gaertn. f. *pyramidalis* Dippel 'Sakari' trees (Alnus site). The sites were established in the summer of 2002 and the trees were planted in the autumn of 2002. At both sites, three different premixed structural soil mixes were applied as a 1-m-deep, 3-m-wide continuous strip (Tilia site) or 15–20-m-long planting pockets for two to four trees (Alnus site). Cast-iron tree gratings 2.25 m^2 in size were used around the trees, and the streets outside the grates were paved over with sand-jointed block pavers. The available soil volume per tree was 45--50 m^3 at the Tilia site and 15--30 m^3 at the Alnus site.

The structural soils consisted approximately 2/3 by volume of stones ranging from 30 to 120 mm in size and 1/3 of fine soil. In soil mix 1, the fine soil was mainly sand, clay and the SOM source was peat. In soil 2, the fine soil was derived from composted sewage sludge mixed with peat, sand and pine bark, but the contribution of peat to the final soil mix SOM could not be determined. In soil 3, the components were fine gravel, sand, clay and leaf compost (peat was not used in the composting process). For soil 1, the initial LOI was 6% and for soil 2 20%, according to their respective manufacturers. For soil 3, the initial LOI was 4.4%, based on the composition and properties of the materials used (7% by volume of leaf compost, 20% of clay with LOI 8.3% and 3% bark mulch). Soils 1 and 2 were commercial mixes, while soil 3 was specially mixed for the study sites.

The transplanted *Tilia* trees were 8--11 cm in diameter-at-breast height (DBH) and *Alnus* 7--11 cm, respectively, and both were balled and burlapped. After transplanting, the trees were not pruned (except for dead and broken branches and shoots growing from rootstocks) until late 2008. Thereafter, the *Tilia* were pruned about annually to achieve the necessary crown lifting. The *Alnus* trees were not crown-lifted, and only branches that were damaged or leaned far out from the columnar crown shape were removed.

Soil water content and temperature measurements

Each soil mix on both streets was instrumented during the establishment with continuously measuring soil moisture sensors (Delta T MLx2 (Delta-T Devices Ltd, Burwell, Cambridge, UK), see Riikonen *et al.*, 2011 for installation details) at depths of 10 and 30 cm and with temperature sensors (resistor type KTY81) at depths of 10, 30 and 60 cm from the surface of the growth media.

Data loggers (DP-158; Envic Oy , Turku, Finland) read the temperature and SW sensors from July 2003 onwards at 1–30-min intervals. At the Alnus site, soil moisture sensors at 30 cm depth were installed in 2005 and data from 10 cm depth was used before that. Soil moisture and soil temperature (T_f) were averaged to 30-min means, and missing data, due mainly to periods of datalogger malfunctions, were gap-filled with linear interpolation. The SW content could only be reliably measured while the T_f was > 0 ºC; periods when ground was frozen were filtered out and gap-filled linearly (Kornelsen et al. 2012) (most missing SW values in 2003: 52%, least missing values in 2006; 7%). The measured volumetric water content was transformed to percentage of soil weight (W_f).

Soil sampling and analysis

Soil samples were collected in the autumn 2005, 2008 and 2011from two pits in each soil mix and site (2x3x2 sampling pits each year, each located in separate planting pocket at the Alnus site, and at least 10 m apart at the Tilia site). The average distance from the nearest tree was 2--3 m, depending on parked cars and other practical considerations. The pavement was removed and a pit with a diameter of 30--50 cm was dug with hand trowels. In 2005, the pits were dug down to depths of 30 cm (sampling depth 0--30 cm, altogether 12 samples), in 2008 to at least 60 cm and to 90 cm where possible (sampling depths 0--30 cm (n=12), 30--60 cm (n=12) and 60--90 cm (n=11), altogether 35 samples), and in 2011 to 60 cm (sampling depths 0--30 cm (n=12) and 30--60 cm (n=12), altogether 24 samples). The rocks (≥30 mm) in the soil mix were separated from the fine soil. All the fine soil excavated from each sampling pit was weighed, thoroughly mixed and a sample of approximately 3 liters was taken from each depth. The sampling pit volume up to each depth was estimated by measuring the volume of vermiculite required to fill the pit. After each measurement the pit was vacuumed empty of any vermiculite and filled back with the original soil material once all measurements were completed.

The soil samples were divided in parts for further analysis. One subsample of approximately 400 g was dried at 105 °C and measured for loose dry bulk density, LOI (550 °C, 2 h) and particle-size distribution (dry sieving and laser diffractometer Coulter LS230; Beckman Coulter Inc., Krefeld, Germany). Another separate subsample was reserved for incubation to estimate the soil CO_2 production rate. The subsamples for incubation were stored at 10 °C for 0--30 h before initiation of the incubation experiment.

Soil incubation

In 2005 and 2011, from all 12 and 24 soil samples collected, respectively, subsamples were incubated and measured for CO_2 production. In 2008, subsamples were collected and incubated from all sites at depths of 0-30 cm and a random sample of one-half of the sites at 30-60 and 60-90 cm. All visible roots were picked out from the soil samples, and eight 20-ml portions from each sample were placed in 120-ml incubation bottles and weighed. The bottles were then flushed with compressed air at atmospheric CO_2 concentration and sealed. The bottles were immediately placed randomly at 5, 10, 15 and 25 °C, two bottles at each temperature. For each soil sample, two bottles were filled with air only and their CO_2 concentration was used as a background level in calculating the results.

After 24 h, the bottles were placed in an ice bucket and gas samples were taken and analysed immediately for CO_2 content with gas chromatography. CO_2 production during the incubation time (R, l CO_2 l^{-1} h^{-1}) was calculated for each subsample, based on the CO_2 concentration in the incubation bottles and measured by gas chromatography (Hewlett-Packard Co., Palo Alto, CA, USA; Jaakkola and Simojoki, 1998), as:

$$R = (CO_{2\,inc} - CO_{2\,ref}) / h \quad \text{Eq. 1}$$

where $CO_{2\,inc}$ (l CO_2 l^{-1}) is the CO_2 content of the incubation bottle after incubation, $CO_{2\,ref}$ (l CO_2 l^{-1}) is the CO_2 content of the air used in filling the bottles prior to incubation and h is the incubation time in hours. Then the C loss (g CO_2 g^{-1} h^{-1}) was calculated as:

$$(R\,V_b / M_s)\,v_m M_{CO_2} \quad \text{Eq. 2}$$

where R is the CO_2 produced in the incubation bottle per hour of incubation (l CO_2 l^{-1}), V_b is the volume of the incubation bottle (l), M_s is the weight of the incubated sample (g), V_m is the gas constant (l mol^{-1}), M_{CO2} is the molar mass of CO_2. C loss in the incubation experiment (g g_{SDW} h^{-1}) (P) was calculated from the CO_2 production:

$$P = g\,CO_2\,g^{-1}\,h^{-1}(M_C / M_{CO2}) \quad \text{Eq. 3}$$

where M_C is the molar mass of C (g mol^{-1}).

Calculation of soil C storage change

The change in soil C stock was calculated as LOI change between the sampling times and from the initial value for different soils and sites (2002--2011). The proportion of C in the LOI used in the calculation was 0.56 (Hoogsteen *et al.,* 2015). Estimates of soil C stock changes were calculated per standard 25 m^3 of soil per tree (Rakennustietosäätiö, 2010) instead of the actual soil volume allocated for each tree at the study sites, to allow easier comparison between sites, soil mixes and tree species. This estimate was compared

with one based on the incubation experiment. Since the first soil sampling for incubation was performed in 2005, the comparison covered the years between 2005 and 2011.

The incubation-based estimate was obtained in two steps. First, we developed an Arrhenius-type model to predict C loss as a function of the measured T_f, SW and LOI content and parameterized it with the soil incubation data (Eq. 4). The three estimated parameters were used in the model (Eq. 5); the intercept (λ), combined LOI and SW response parameter α and parameter β describing the temperature response. After the testing of year-, soil- and site- specific datasets, the least number of separate models that presented no heteroscedasticity problems were chosen. As a result, the model was fitted separately for each soil type and site (n = 40 in each combination):

$$\ln(P) = \lambda + (\propto \ln(O)\ln(W)) + \beta T, \quad \text{Eq. 4}$$

where O is the LOI of the soil sample (% of dry weight), W (% of dry weight) is the SW of the soil sample and λ, α and β were the fitted parameters. The model was then applied to predict the soil C loss, based on the measured T_f and water content at the studied sites:

$$C\ loss = exp(\lambda + (\propto (lnO_e)\ lnW_f) + \beta T_f + (RMSE/2)^2), \quad \text{Eq. 5}$$

where λ, α and β are the parameters fitted in the preceding step, W_f is the measured SW content (% weight) and T_f is soil temperature (°C) hourly averages. The correction for unbiasedness (Baskerville, 1972) was calculated from the root-mean-squared error (RMSE) of the parameter estimation. The initial LOI ($O_{e\ ini}$) at the starting point of the model was generated with a combined two first-order kinetic model (Bernal *et al.*, 1998), based on the initial LOI and measurements from the soil samples taken in 2005, 2008 and 2011 as:

$$O_{e\ ini} = LOI_{2002} - (LOI_{2002}\ (1 - \exp(-K_r t_m)) + S\ (1 - \exp(-K_s t_m))) \quad \text{Eq. 6}$$

where LOI_{2002} is the LOI of the soil at T_0, t_m is time after establishment (in months), and K_s, K_r and S are the fitted parameters. After the first time step of the incubation model, subsequent O_e values were obtained dynamically from the periodic CO_2 production estimates calculated with the model. Half-hourly measurements of T_f at depths of 10 cm were used for depths of 0--20 cm, at 30 cm for depths of 20--45 cm and at 60 cm for depths of 45--100 cm. The SW measured at 10 cm was used for the 0–20-cm layer and measurement from 30 cm for lower levels.

The incubation model based total C loss for each 25 m^3 of tree soil was calculated as:

$$Total\ C_{inc}\ loss = C_{tot}\ \sigma\ v\ 25 \quad \text{Eq. 7}$$

where σ is the loose soil bulk density (kg m^{-3}) and v is the proportion of fine soil in the structural soil (m^3 m^{-3}).

The LOI based C loss (kg per 25 m^3 of tree soil) was separately calculated, based on the LOI change between the soil construction and year 2011:

$$Total\ C_{LOI}\ loss = ((L_{2002} c\ \sigma\ v) - (L_{2011}\ c\ \sigma\ v))\ 25 \quad \text{Eq. 8}$$

where L_{2011} is the measured loss on ignition (g g^{-1}) measured in 2011, c is estimated proportion of C in the *LOI* (0.56; Hoogsteen *et al.*, 2015), σ is loose soil bulk density (kg m^{-3}) and v is proportion of fine soil in the structural soil (m^3 m^{-3}).

Tree biomass measurements

The development of tree aboveground woody biomass (ABW) at the study sites was calculated from measurements made of same 6--12 trees per species in August 2005, 2008 and 2011, and leaf biomass from measurements taken in all years from 2004 to 2011. The selection of observed trees (a combination of systematic and random sampling, each soil type at site was equally represented) and leaf area measurements are described in Riikonen *et al.* (2011). Tree trunk diameter was measured at soil surface, at height of 130 cm and above all first-degree branches. The height and diameter of each first-degree branch were recorded. The living crown was divided horizontally into three equally long sections, and leaf samples were taken from the median diameter branch of each crown section for leaf area and dry weight measurements. A relationship was fitted between the branch basal area and leaf area and used to predict the leaf biomass for all branches within the tree crown (Riikonen *et al.* 2011).

The branch woody biomass (WB) for *Tilia* was based on 206 and for *Alnus* 46 branches, bulked per species, pruned from the study site trees. A power function was fitted to estimate the branch dry biomass, based on its diameter:

$$M_{wb} = \delta\ D_b{}^{\mu} \quad \text{Eq. 9}$$

Here, M_{wb} is the woody biomass of the branch (kg), D_b is branch diameter (mm), and δ and μ were the estimated parameters. The biomass of all branches in each tree was calculated based on the branch diameters measured. The tree trunk volume was calculated from the trunk diameter measurements as stacked cylinders and converted to biomass with specific gravity for *Tilia americana* L. (0.40) and *Alnus rubra* Bong. (0.43) (Alden, 1995).

Living tree roots encountered in the sampling pits during soil sampling in 2008 and 2011 were collected for root biomass estimation, washed and weighed (R_f). Additionally, small roots that could not be separated in

the field were separated from the dried soil sample, weighed and calculated per kg of soil (R_d). The total dry root biomass for each sampling pit was calculated, based on the roots collected in the soil sampling as:

$$Root\ DW = ((R_f j) + (R_d M_p))/V_p \quad \text{Eq. 10}$$

where j is the root dry-to-fresh weight ratio, M_p is total mass of the collected fine soil sample from the sampling pit (kg) and V_p is volume of the pit (m^3).

The total tree root dry biomass (g m^{-3}) was estimated from roots in the collected soil samples, assuming the root density was equal to the average of all sampling pits at each site. This value was assumed to represent root density for the entire root system. It is estimated that the root system covers an area with a radius of 38 x the DBH of the tree in question (Day *et al.*, 2010), which exceeded the tree soil dimensions already in 2008; thus it was assumed that roots had grown throughout the entire soil volume. We assumed that the root density in the root ball at planting time was equal to the value in 2011 and calculated an estimate of root biomass at planting, based on the root ball volume (diameter 100 cm).

C stocks in branch prunings and leaf litter

The contribution of exported biomass fractions (tree litter C and pruned branches) to tree C sequestration was estimated with an exponential decay function (Olson, 1963), to account for these non-living compartments of tree sequestrated C at a given time. The function was applied for each fallen leaf and pruned branch cohort separately for each year:

$$M_r = M_0 \exp(-kt) \quad \text{Eq. 11}$$

where M_r is the remaining leaf or branch biomass, M_0 is leaf or branch biomass (kg) at t_0 (years), k is decay factor (*Tilia* leaves: 0.24, Hobbie *et al.*, 2006, *Alnus* leaves: 0.6, Dilly and Munch, 1996, and branches of both species: 0.22, Perruchould *et al.*, 1999) and t is years passed. We assumed that the C in the leaves was initially equal to the total C content of that year's leaf cohort and was lost at a rate equal to the total mass loss. The WB removed by pruning was measured for one half of the trees, selected randomly, at each pruning.

Prediction of tree DBH, biomass and C

To estimate the compensation point (the number of years after planting required for the tree C capture to reach the sum of the soil C loss in the first decade after planting, not accounting for possible soil C changes after first 10 years) for the case study trees, we needed to predict the long-term tree growth and biomass accumulation. For this we needed models that predict both the tree growth over time and how the biomass is distributed among the various tree parts. We compared existing DBH growth models (we found two for *Tilia*, but only one for *Alnus*) (Table 2) with observations from sample trees that we collected for this

purpose. The biomass and C stock predictions were based on BEs (Table 2, Appendix), assessed and selected by comparing the BE predictions with the measured values explained above.

We collected the DBH values from 19 *Tilia* sp. street tree plantings in Helsinki with known planting year and at least six trees still remaining of the originally planted. We used either the latest DBH data from the City of Helsinki street tree database or measured at least six randomly chosen trees per planting in 2012—2014. Only two known *Alnus glutinosa* f. *pyramidalis* plantings in Helsinki are older than the case study site. Two datasets of earlier DBH measurements were available for one of the plantings and one set for the other, and additionally, all trees were measured for DBH in 2014: altogether, the resulting dataset on *Alnus* DBH had 5 data points (average DBH at given age). For Tilia, from 6 to 68, on average 22 trees were measured to attain the average DBH for a planting site, and for Alnus, one planting had 22 and the other, 91 trees.

To predict the C content of the pruned branches, we applied a pruning regime the City of Helsinki aims for with street trees: the trees were pruned 1, 3, 5, 7, 10, 15 and 20 years after planting and thereafter every 10 years. In the first 20 years, we applied a pruning of 25% and thereafter 15% of the branch biomass at each pruning.

In all conversions from woody biomass to C, we used a 45% C content for both species. We used C content of 47.6% for *Tilia* (Niinemets, 1999) and 45.8% for *Alnus* (Browaldh, 1997) leaves, respectively. For purposes of predicting future C stocks of tree planting, we assumed there was no further soil C stock change after the first 10 years.

Statistical analysis

The mean LOI change-based C loss for each site and soil type, and similarly, the measured and BE-predicted tree biomass compartments for individual trees in 2011, were each log-transformed and compared using 2-sided Tukey's test. The incubation-based C loss prediction model parameters were estimated with SAS procedure MODEL (linear regression, Eq. 4), and the residuals were assessed with White's heteroscedasticity test. The relationship between sample branch diameter and biomass in the tree biomass measurements was determined with nonlinear regression (Eq. 9). DBH growth models for Tilia were compared to measurements by calculating RMSE and bias and testing the significance of the latter with t-test. The p value required for significant difference and/or effect was set at ≤ 0.05 for all statistical analyses.

Results

Soil C storage change

The soil C stock at the time of establishment was approximately 400, 900 and 250 kg of C per tree in the 25 m^3 of tree soil in soils 1, 2 and 3, respectively. The average loss of C in 2002—2011 across the soils was approximately 290 kg per 25 m^3, as calculated from the LOI change (Figure 1). The highest proportional LOI loss in comparison to the original LOI was measured in soil 3; only about 1/3 remained in 2011 (Figure 2). The peat-based soil 1 maintained its LOI best, retaining on average, 56% of the original LOI value, with C loss estimates of 100-250 kg per 25 m^3.

Based on the incubation model, the average annual soil C loss per m^3 in 2005—2011 was 0.41 kg, resulting in a slightly lower C loss over time in all soils than in the LOI loss-based estimate (Figure 3). Model parameter β describing the temperature response was highest in soil 2, but parameter α combining the effects of LOI and SW was highest in soil 3 (Table 3). Periods of extrapolation to T_f values outside the incubation experiment range accounted for 17.5% of the C loss predicted by the model. The gap-filling in the SW data produced only 0.54% of the C loss estimate.

The particle-size distributions showed that in 2002, 83% of the samples by weight on average were < 2 mm, but in 2011 the corresponding value was only 58%. Based on laser diffraction analysis, the clay content of all samples was < 1%.

Tree biomass C

Based on biomass measurements, an average *Tilia* tree showed approximately 25 kg of C in its WB in 2011 (Table 4) and sequestered approximately 18 kg C in total WB per tree in 2003—2011. The WB of an average *Alnus* tree had a stock of 37.5 kg of C in 2011. Including the C in the roots, C sequestration in the *Alnus* WB in 2003—2011 showed a total of 32 kg.

The branches used to predict *Tilia* and *Alnus* branch biomass, based on branch diameter, showed fairly good correlation between branch diameter and biomass; the R^2 values were 0.93 and 0.91, respectively. The estimated leaf and litter contribution to the total C stock was high, especially for *Tilia*: 7.3 kg per tree in 2011 (Table 4). The prunings contributed relatively little to the C stock in either species. Adding the C estimated to remain in the leaf litter and pruned branches to the tree C sequestration estimates brought the sequestered C in 2011 to 26 kg per *Tilia* tree and for *Alnus* to 38 kg per tree.

There were significant differences in all aboveground biomass compartments measured between the tree species, but only two of the seven BEs tested (Appendix) produced significantly different total or ABW estimates from the measured biomass (Table 5). However, all of the BEs showed significantly higher stem biomass and lower branch and leaf biomass in comparison to the measurements.

Tree C sequestration vs. soil C storage change

Based on the estimates of soil C loss and tree C sequestration, the *Alnus* trees sequestered 10—15% of the mean soil C loss by 2011, while the *Tilia* trees achieved some 10% sequestration in comparison to the mean C losses from the tree soils, but the uncertainties in both soil C loss and tree C sequestration remain considerable. In the highest C loss soil, less than 10% of the C loss was sequestered by the trees, while in comparison to the lowest loss soil and site, the *Alnus* trees approached 40% sequestration in 2011. These percentages include the C in leaf litter and prunings, as estimated for 2011. Considering only the peat-based soil 1, *Alnus* trees had sequestered about 30 kg of C, while the C loss from soil 1 was estimated as 170 kg (average over the two sites, Figure 1).

Of the available DBH growth models for long-term biomass accumulation estimation, DBH model 2 coincided better with the measurements (Figure 6) with a relative RMSE of 20% and a bias of 0.7 cm, while DBH model 1 showed tree sizes similar to those at the best sites measured in Helsinki with relative RMSE of 28% and a bias of -7.4 cm. Neither model predictions were significantly different from measurements however. The data collected from the *Alnus* plantings in Helsinki were too limited to assess the applicability of the prediction.

If *Tilia* growth followed prediction 2, the estimated C capture in the woody biomass of the *Tilia* trees would reach the mean soil C loss of the first decade about 55 years after transplanting (Figure 7) and the smallest soil C loss in this study (Soil 1 at the Alnus site, Figure 1) in about 30 years. The prediction for *Alnus* was more uncertain still, but it showed a biomass accumulation curve very similar to that in prediction 2 for *Tilia*, with *Alnu*s some 5 years ahead (Figure 7). Taking the estimated litter and prunings C stock into account moved the previous compensation point estimates forward by 3 years for *Tilia*, but in the *Alnus* C stock, the litter and prunings had little effect.

Discussion

Soil C storage change

High C losses from tree planting soils were seen soon after soil construction. In the two compost-containing soil mixes studied, the LOI was halved in less than 10 years and most of the LOI loss took place before the first soil sampling in 2005. In addition to our study, rapid SOM decline has been demonstrated for other compost-derived growth media, especially when the compost was not sufficiently mature (e.g. Bernal et al., 1998; Sanchez-Monedero et al., 2004; Vidal-Beaudet et al., 2012). In the entirely peat-based soil mix 1, the LOI loss was not as high, consistent with studies of greenhouse growth media (e.g. Prasad and O'Shea, 1997; Prasad and Maher, 2003).

409 The incubation model gave lower values for C loss than the LOI loss measured, and the parameters for the
410 model differed clearly between sites. Sampling structural soil is complicated by the presence of load-bearing
411 stones, and consequently the samples incubated were heavily disturbed. This may have affected the resulting
412 CO_2 production rates. The absence of active tree roots may also have had a suppressing effect on SOM
413 mineralization (Linden *et al.*, 2014). The estimated parameters showed the highest temperature response in
414 the highest LOI soil mix, likely due to the higher amount of decomposable organic material. The water
415 content and LOI response were more varied, probably because the ranges of these variables did not entirely
416 overlap in the various soils and sites. Overall, the number of samples per soil and site was rather low, and the
417 site differences may be exaggerated. On the other hand, the differences between sites, such as the planted
418 tree species (e.g. Bomberg and Timonen 2009) or soil water status (e.g. Williams and Rice 2007) in long
419 term may have led to different soil microbial community and SOM decomposition on initially similar soil
420 mixes.
421
422 Based on the change in particle-size distribution and visual inspection of soil samples, an increase in coarse
423 material was seen in the fine fraction after soil construction, originating from breakage of stones in the
424 structural soil during handling and tamping. Assuming the change in finer fractions was similar to that seen
425 in the coarse fraction, an addition of some 25% in volume can be roughly estimated. This would reduce the
426 LOI loss estimates by 20%. From Figure 7 we can assess that such an error has a relatively small impact on
427 the compensation point estimates. The clay content of the samples was measured with the laser diffraction
428 method, which gives lower values for clay content than does the pipette method (2—3 times less; Taubner *et*
429 *al.*, 2009). The clay content was low nevertheless, and no clay correction was applied in the C stock
430 calculation.
431
432 At the time of the site establishment, the LOI of the growth media was not separately measured. The initial
433 LOI values were bulk values from the growth media trade descriptions (soils 1 and 2) and the LOI values of
434 the organic matter used in the soil mix for soil 3 (specially prepared > 200-m^3 mix).The LOI change
435 estimates were net C changes in the soil, i.e. the possible contribution of root litter to the soil C stock was
436 included in the net effect calculation; it would add SOM to the soil and thus reduce the soil C loss observed.
437 The agreement seen between the LOI and incubation-based model, implies that the uncertainties were
438 probably not major. However, the estimates of LOI half-life are especially sensitive to the types of
439 uncertainties present in the data.
440
441 *Tree C sequestration*
442 For the relatively small trees measured in this study, the differences between the various BEs for total
443 aboveground biomass (AB) or total ABW and the measured biomass were fairly small for five of the seven
444 equations. The only model for urban *Tilia* trees (McHale *et al.*, 2009) gave much higher values than the other

BEs, but the authors noted that the trees were irrigated and fertilized. For *Alnus*, BE 5a deviated furthest from the measured biomass. The source of the equation also gave separate BEs for the stem and branches (BEs 5b and 5c); summing these up resulted in better estimates (Table 5).

While the total aboveground BEs performed fairly well, the equations for the various AB compartments – leaves, branches and trunk – did not. In *Alnus*, this may have been due to the different crown form (f. *pyramidalis*) in comparison to the BEs and the lack of large branches in the biomass samples. For *Tilia*, however, the branch biomass was more likely under- than overestimated, because large branches in the biomass samples were often reduction-pruned previously. The differences between the measurements and BE predictions imply that in *Tilia* street trees, the biomass distribution within the crown likely differs from that of *Tilia* in traditional forests. Similar findings were recently obtained for three other species in open sites (Zhou *et al.*, 2014); stem biomass was low and branch biomass was high, but total aboveground biomass agreed somewhat with traditional forest based equations. More data needs to be gathered however, as our study only concerns a small number of trees from two species.

The leaf biomass of the trees measured was also consistently higher than the BEs predicted. In our data, the *Tilia* trees annually invested approximately 40--50% and *Alnus* 30--40% of the total aboveground C increase into leaves, which is on the high side but within the range for similarly aged stands in allocation studies reviewed by Litton et al. (2007). The roots of the *Tilia* trees appeared to have relatively more C stock than the roots of the *Alnus* trees. This may have resulted from the uncertainties in root sampling however, since the variation between samples was high. Based on the literature, belowground biomass can be estimated to be 23% of the AB (Chojnacky *et al.*, 2014), giving estimates surprisingly close to the measurements, so while our data is very uncertain, using literature based values instead would have had little effect on the overall results.

The contribution of litter and prunings to overall tree C sequestration appeared large for the still rather small case study trees. However, the literature-based decay factors may not have performed well under our conditions; the contradictory results concerning urban vs. rural litter decomposition rates (Pouyat *et al.*, 1997; Pouyat and Carreiro, 2003; Nikula *et al.*, 2010; Dorendorf *et al.*, 2015) indicate that the processes related to urban litter are not yet well understood. When the soil surface is sealed, the aboveground litter is likely lost from the tree-soil system, warranting leaving it out of the C sequestration estimates. However, the C input in leaves was quite high in the case study trees, indicating that improving leaf C retention and longevity may be one key factor in improving urban tree C sequestration.

Tree C sequestration vs. soil C storage change

In tree C sequestration, the favourable growth rate of trees is critical, because biomass is a function of tree size, and tree size growth is dependent on the growing conditions. The uncertainty in tree C sequestration

predictions in general is thus mainly linked with the difficulty in predicting tree growth rate and mortality (Strohbach *et al.*, 2012); this is easily seen in the DBH data of existing *Tilia* plantings in Helsinki (Figure 6). Identifying the remaining original trees and the planting year data was not easy in all cases, which may explain some of the variation. Since the BEs appeared to underestimate branch biomass, the biomass in the prunings was likely also underestimated. This may have been compensated for somewhat, because the pruned branch biomass was not removed from the predicted tree biomass.

In forest ecosystems, trees accumulate C as they grow, but soil C dynamics are dependent on litter input and decomposition, which can be affected by disturbances such as tree felling. Research on traditional forest soil C stock temporal dynamics is often contradictory and confounded by the various initial states (e.g. fire, harvest or plantation) of the system (Yang *et al.*, 2011). Similarly, planting of urban trees is usually preceded by a land-use change, making it difficult to draw parallels with native ecosystems; however, there are some studies showing recovery of urban soil C stocks over time (e.g. Golubiewski, 2006; Bae and Ryu, 2015; Setälä et al. 2016). In managed traditional forests of Finland, soil C stock decreases for approximately 20 years after clearcutting (Peltoniemi *et al.*, 2004) but net C sequestration in the ecosystem can be reached after some 10--15 years (Kolari *et al.*, 2004). The most positive compensation point estimates in this study were similar, yet we must conclude that the often stated expected street tree lifespan of 20--30 years (Roman and Scatena, 2011) appears unlikely to bring tree C sequestration benefits in soil mixes in the study, even if only the entirely peat-originating loss of C on soil 1 is considered accountable. Fortunately, the annual C capture of young trees can be expected to improve for several decades (Pregitzer and Euskirchen, 2004).

It is also likely that the belowground litter C input will increase the soil C stock over time. A pattern of initial soil C loss, followed by soil C stock increase, in conjunction with change to urban land use has been emerging in the USA (Pataki et al. 2006). In our study, soil C loss was high in the beginning, but stabilized after the first few years (Figure 2); it is possible that soil C will start to increase as time passes, paralleling this type of development. The estimates of compensation point timing are thus very uncertain in relation to possible soil C stock increase over time and must be considered as only suggestions for the minimum tree life expectancy needed to produce net C sequestration by planting street trees in artificial growth media. The design of our study was not particularly well suited for generalization of the results but rather it demonstrated possible long term C dynamics in urban planting schemes. There are many C expenses related to street trees that we are not accounting for, such as C cost of maintenance (McPherson *et al.* 2015) and cast iron tree grates and trunk guards. Despite these shortcomings, the range of results indicate that the soil C dynamics need attention in urban tree C sequestration estimates overall.

Both peat and compost derived organic matter are currently used in growth media; the latter appears to be the more sustainable choice, although the sustainability determined is greatly dependent on the assessment

method chosen (Defra, 2009). The use of peat in composting process adds to the difficulty of choosing suitable tree soil. While high SOM contents have long been considered beneficial for soil quality (Reeves, 1997), perhaps the amount of SOM used in tree soils could be optimized more carefully. Biochar and other e.g. clay-stabilized C soil additives appear very attractive new options for C sources in artificial growth media due to their relative stability in comparison to compost (Bolan *et al*. 2012, Ameloot *et al*. 2013).

However, the C cost of tree planting, or the possible lack of C-related overall benefits, should not discourage people from planting street trees. C sequestration is only one of the ecosystem services provided by trees, and its value is usually estimated to be small in comparison to storm water management, property value and energy-saving benefits, not to mention recreational and cultural values and human health benefits (McPherson *et al.*, 2005). Rather, these results should encourage more investment and interest in tree planning, establishment and maintenance to ensure improvements in urban tree lifespan and eventual tree C sequestration. Overall, using local soils and less and lighter infrastructure in tree plantings would likely lead to higher net C sequestration. Unfortunately, these appear to be a rare option in the midst of efforts towards more efficient urban land use.

Conclusions

In considering the C sequestration capacity of street trees, the initial C losses from peat-containing soils after site establishment should not be overlooked. Due to the exponential nature of the increase in tree C stock over time, tree planting-related C expenses are best compensated for with a long tree life expectancy. This is best achieved with adequate consideration of tree growth requirements over their entire life cycle. Compensating for the smallest measured single tree soil C stock loss of 100 kg by C sequestration of the tree in this case study was predicted to require at least 30 years. This exceeds current estimates of average street tree lifespan. Especially the estimates of tree root C stock and soil C development in the future were uncertain however.

Further research on biomass distribution within urban trees, both above and under ground, is needed to improve the accuracy of urban C stock and C stock change estimates. The use of traditional forest based BEs for urban trees may lead to errors when specific AB compartments are estimated. This causes further problems in assessment of the effects of leaf litter and prunings in urban tree C sequestration. Cutting the C expenses of the tree planting and focusing on the longevity of pruning and leaf litter C may aid in balancing tree-related C effects.

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770 Table 1. A summary of major measurements performed for the case study (Tilia and Alnus sites in 2003-
771 2011), and numbers of samples taken and/or analyzed, at Tilia and Alnus sites. T= temperature (°C) D=
772 diameter (mm), BM= biomass (g).
773

	Soils				Trees			
Site and measu-rement	T_f (every 30 min)	W_f (every 30 min)	LOI, loose BD, particle size distribution	T, SWC, LOI response of CO_2 production	Trunk and branch D	Leaf area	Prunings (BM and D/BM ratio)	Root BM
Tilia site	2003-2011, 3 sensors at 30, 60, and 90 cm depth, respectively	2003-2011, 3 sensors at 30 and 60 cm depth, respectively	2005 (n=6), 2008 (n=15), 2011 (n=12)	2005 (n=6), 2008 (n=12), 2011 (n=12)	2005 (n=8), 2008 (n=7), 2011 (n=6)	2004-2011 (n=6-12 trees, 3 branches per tree)	Every time trees were pruned. (2008-2011, n=7)	2005, 2008, 2011 (n=6)
Alnus site	2003-2011, 3 sensors at 30, 60, and 90 cm depth, respectively	2003-2011, 3 sensors at 30 and 60 cm depth, respectively	2005 (n=6), 2008 (n=15), 2011 (n=12)	2005 (n=6), 2008 (n=12), 2011 (n=12)	2005 (n=10), 2008 (n=7), 2011 (n=6)	2004-2011 (n=6-12 trees, 3 branches per tree)	Whenever pruned (2010, n=11)	2005, 2008, 2011 (n=6)

774
775 Table 2. Details of the DBH growth and biomass C accumulation forecast models. BE s (biomass equations)
776 referred can be found in the Appendix.

	Forecast 1 for *Tilia*	Forecast 2 for *Tilia*	Forecast 1 for *Alnus*
DBH growth forecast			
	Street trees (Larsen and Kristoffersen, 2002)	Traditional forest trees (Yield class III, Böckmann, 1990)	Traditional forest trees (Yield class III, Schober, 1987)
Biomass accumulation *by compartment*			
Aboveground woody		BE 1	BE 7a minus leaves (7b)

Roots	23% of aboveground woody biomass (Chojnacky *et al.*, 2014)	23% of aboveground woody biomass (Chojnacky *et al.*, 2014)
Litter		
Leaf	BE 3b	BE 7b
Branch (prunings)	BE 3c	BE 5b
Decay function; parameters (leaves, branches)	Olson 1963; 0.24 (Hobbie *et al.*, 2006), 0.22 (Perruchould *et al.*, 1999)	Olson 1963; 0.60 (Dilly and Munch, 1996), 0.22 (Perruchould *et al.*, 1999)

Table 3. Parameter values ± SD, R^2 and White's heteroscedasticity test p-value for soil and site specific P prediction models (Eq. 4). The parameters λ (intercept), α (SW and LOI parameter), and β (temperature response parameter) were used to predict C loss (Eq. 5).

Site	Soil mix	λ	α	β	Adj. R^2	White's test p-value
Tilia	1	-18.02 ± 0.68	0.083 ± 0.013	0.08 ± 0.149	0.51	0.19
	2	-19.43 ± 0.64	0.083 ± 0.011	0.44 ± 0.078	0.68	0.57
	3	-17.81 ± 0.17	0.065 ± 0.008	0.34 ± 0.083	0.65	0.08
Alnus	1	-19.16 ± 0.21	0.084 ± 0.011	0.48 ± 0.083	0.71	0.43
	2	-18.70 ± 0.26	0.108 ± 0.008	0.16 ± 0.034	0.84	0.82
	3	-18.91 ± 0.14	0.090 ± 0.006	0.67 ± 0.142	0.85	0.50

Table 4. Estimated C in the *Tilia* and *Alnus* aboveground stem, branch, roots and total C in woody biomass; cumulative original C in leaves and pruned branches and remaining C in pruned branches and leaf litter during the period examined from 2002 to 2011 (kg per tree ± SD when estimable). The remaining C in the leaves and branches was calculated for each cohort separately and summed up. ABW = aboveground woody biomass.

	Year	Stem	Branches	ABW	Roots	Total woody	Leaf C remaining	Prunings C remaining	Total litter + prunings	Sum
Tilia	At planting	n/a	n/a	6.7*	0.2	6.9*	0	0	0	6.9*
	2005	2.5 ± 0.4	2.3 ± 0.6	4.9 ± 0.9	n/a	6.3**	1.5	0	1.5	7.8**
	2008	5.4 ± 2.1	4.1 ± 2.1	9.5 ± 4.0	3.4 ±5.2	12.9	3.9	0	3.9	16.8
	2011	8.4 ± 4.1	8.2 ± 2.8	16.6 ± 6.7	8.5 ±9.7	25.1	7.3	0.6	7.9	33.0
	C stock increase	n/a	n/a	9.9	8.3	18.2	7.3	0.6	7.9	26.1
Alnus	At planting	n/a	n/a	5.6*	0.1	5.7†	0	0	0	5.7*
	2005	4.2 ± 0.8	2.9 ±0.6	7.1 ± 1.3	n/a	9.2**	2.1	0	2.1	11.3**
	2008	8.2 ± 1.3	6.9 ± 1.5	15.1 ± 1.0	3.1 ± 4.1	16.5	3.8	0	3.8	22.0
	2011	17.6 ± 2.8	12.6 ± 2.7	30.2 ± 4.3	7.3 ± 7.1	36.0	6.3	0.1	6.3	43.9
	C stock increase	n/a	n/a	24.6	7.2	31.8	6.3	0.1	6.3	38.2

* ABW with BE 1.

† ABW with BE 5a

**Root biomass estimated as 23% of ABW (Chojnacky *et al.*, 2014) added to total woody biomass

Table 4. Biomass of the various tree compartments as measured (kg per tree ± SD, n = 6—12) and predicted from the DBH with biomass equations (average of the trees included in the biomass measurements) for each taxon in 2011. The number and letter in superscript indicate the equation used (Appendix). ABW = aboveground woody biomass, AB = aboveground biomass, including leaves. * denotes that the biomass model prediction is significantly different from the measurement results (Tukey's 2-sided t-test).

	Stem	Branch	Leaf	ABW	AB
Tilia meas.	18.6 ± 9.1	18.2 ± 6.2	4.5 ±1.0	36.8 ± 14.9	41.3 ± 15.3
Tilia pred.	36.1 ± 13.5* [3d + 3e]	7.5 ± 3.2* [3d]	1.4 ± 0.5* [3b]	43.2 ± 17.0 [1], 79.3 ± 26.4* [4]	47.8 ± 19.4 [2] 42.0 ± 15.9 [3a]
Alnus meas.	39.1 ± 6.3	28.0 ± 5.9	6.8 ±1.3	67.1 ± 9.6	73.9 ± 10.8
Alnus pred.	61.0 ± 7.4* [5c] 69.3 ± 6.4* [6d] 60.7 ± 6.5* [7d]	12.8 ± 1.6* [5b] 5.3 ± 0.7* [6c], 7.8 ± 1.3* [7c]	1.0 ± 0.1* [6b] 1.8 ± 0.1* [7b]	48.0 ± 5.6* [5a] 73.7 ± 8.9 [5b+c] 73.5 ± 7.3 [6a-b] 69.1 ± 7.8 [7a-b]	74.5 ± 7.4 [6a] 70.9 ± 8.0 [7a]

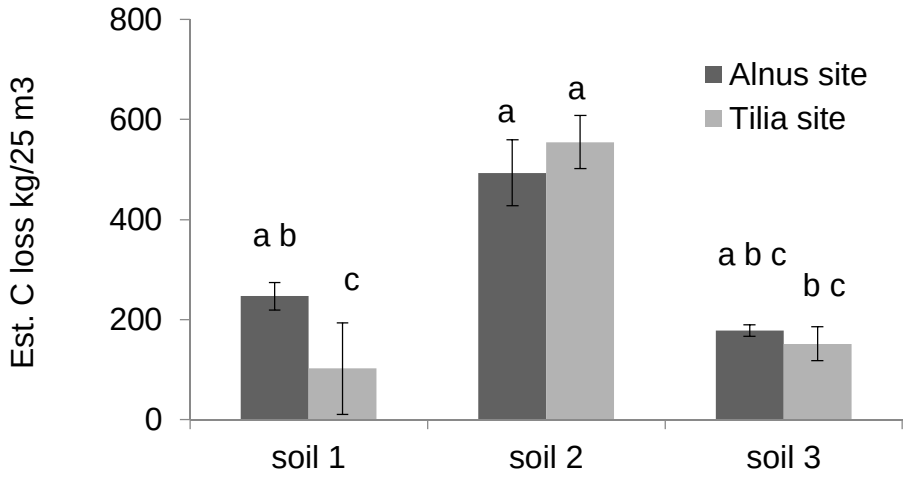


Figure 1. Estimates of cumulative C loss (± SD) estimated per the 25 m^3 of the three different tested tree soils from 2002 to 2011, based on the soil loss-on-ignition change. The mean values indicated with the same letter do not differ significantly.

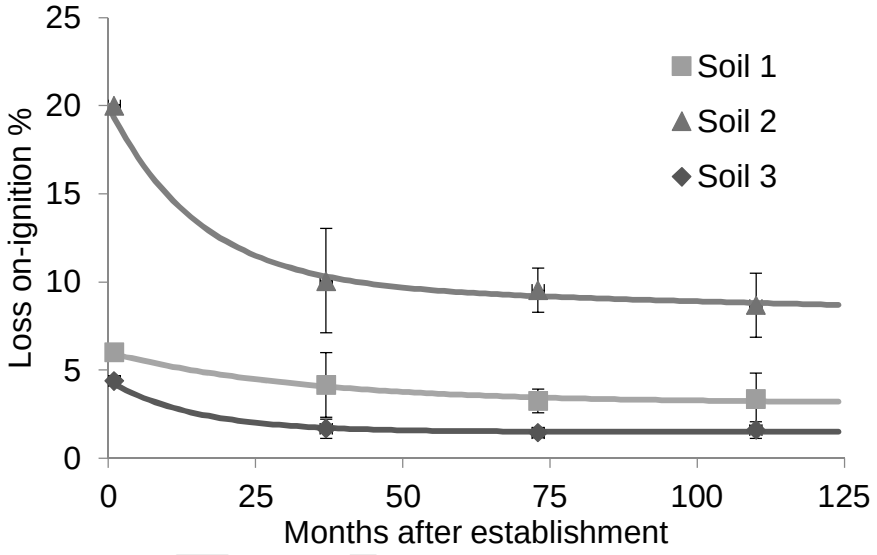


Figure 2. Initial and measured average loss-on-ignition (± SD) for each soil mix fine soil fraction at each sampling time (markers), and estimates for LOI between LOI sampling from the incubation model (O_e) (lines, Eq. 6) for each month after establishment.

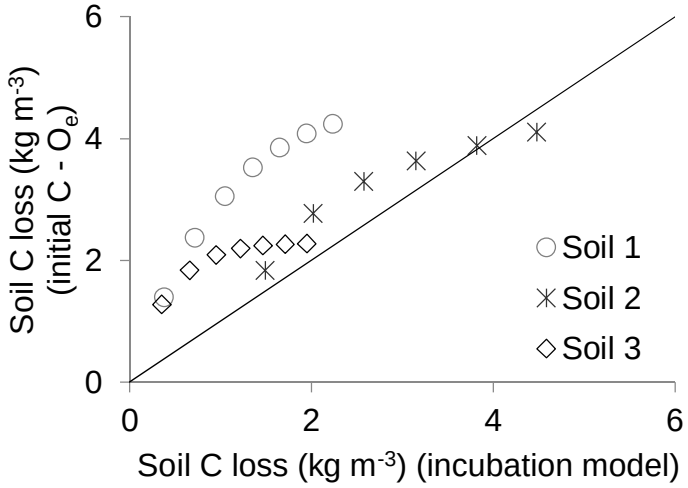


Figure 3. Cumulative C loss in 2006–2011, based on the O_e value (Eq. 6; Figure 2) on the X axis and the incubation model on the Y axis, with early 2005 as the starting point.

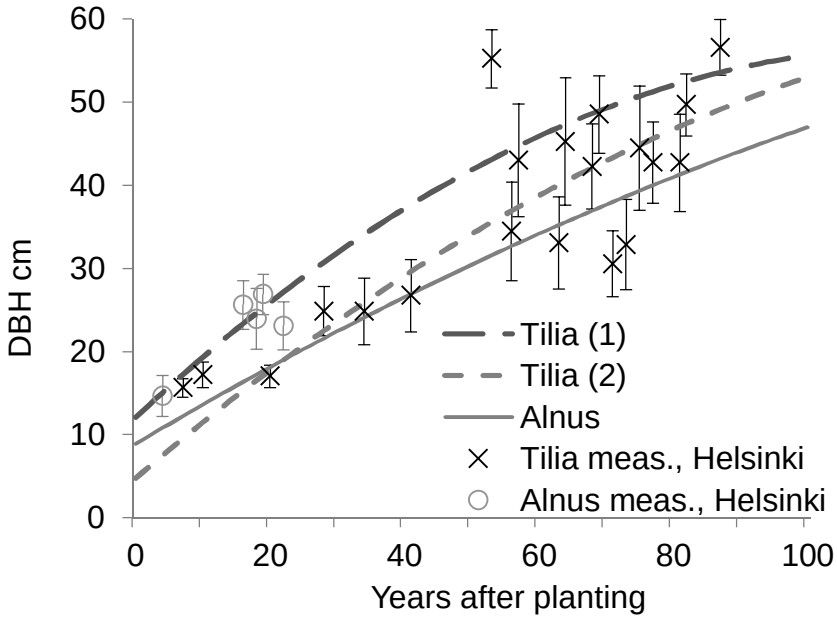
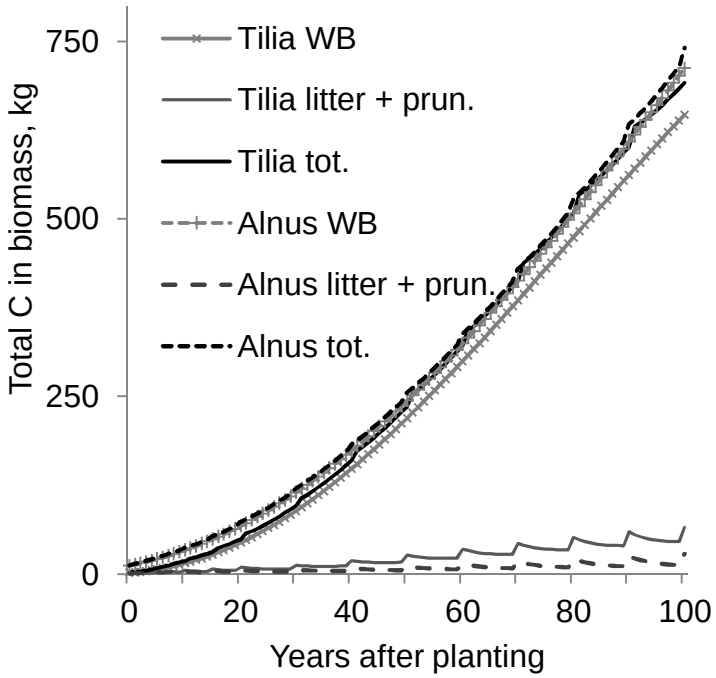


Figure 6. Measured and predicted DBHs for *Tilia* and *Alnus*. The DBH predictions, two for *Tilia* and one for *Alnus*, are based on literature (see Table 2). The crosses mark the average DBHs measured from the street tree *Tilia* plantings in Helsinki (± standard deviation, SD) plotted against the number of years after planting. The average DBH (± SD) of the two known older *Alnus glutinosa* f. *pyramidalis* plantings in Helsinki, for one at two and for the other at three available time points, are marked with circles. Average predicted DBH growth rates at 0-20 years, 0.65, 0.69, and 0.46 cm yr^{-1}, at 20-40 years, 0.57, 0.56 and 0.42 cm yr^{-1}, and >40 years, 0.40, 0.31 and 0.34 cm yr^{-1} for Tilia predictions 1 and 2, and Alnus prediction, respectively.



826
827 Figure 7. Predictions for tree C sequestration in woody biomass (WB, including roots), litter and pruned
828 branches (litter + prun.) and the sum of the previous (tot.) at 0—100 years after planting. The *Tilia* C stock
829 was calculated based on DBH prediction 2 (Figure 6).
830
831

Appendix.

Parameters and references for the allometric equations (BEs) for the total AB and branch, trunk and leaf biomass used in the calculations in this study. Equations 1--4 are for *Tilia* sp. and 5--7 for *Alnus glutinosa*. The DBH ranges for which the equation was developed are listed. The equation form is $M = aD^b$, where M = biomass (kg) and D = DBH, unless noted otherwise.

	Allom. equation no.	Parameter *a*	parameter *b*	Biomass compartment	DBH range and unit	Reference
Tilia	1	-5.49	2.45	woody abovegr.*	3--15 cm	Bunce, 1968†
	2	0.062	2.53	total abovegr.	5--50 cm	Brenneman, 1978, ref. Ter-Mikaelian and Korzukhin, 1997 ††
	3a	0.087	2.35	total abovegr.	4--47 cm	Perala and Alban, 1994††
	3b	4.90×10^{-3}	2.09	Leaves ‡		
	3c	6.59×10^{-3}	2.68	branches		
	3d	4.99×10^{-2}	2.40	stem wood		
	3e	4.32×10^{-2}	2.03	stem bark		
	4	9.40×10^{-2}	2.04	woody abovegr.**	n/a, cm	McHale *et al.*, 2009†
Alnus	5a	8.60×10^{-2}	2.35	woody abovegr.	n/a, cm	Hughes, 1971
	5b	1.47×10^{-2}	2.52	branches		
	5c	8.42×10^{-2}	2.45	stem		
	6a	3.09×10^{-3}	2.02	total abovegr.	120--280 mm	Johansson, 1999
	6b	3×10^{-6}	2.55	leaves		
	6c	3×10^{-6}	2.88	branches		
	6d	5.61×10^{-3}	1.89	stem		
	7a	7.90×10^{-4}	2.29	total abovegr.	20--170 mm	Johansson, 2000
	7b	2.39×10^{-3}	1.33	leaves		
	7c	6×10^{-7}	3.28	branches		
	7d	1.19×10^{-3}	2.17	stem		

* $\ln M = a + b (\ln G)$, G = girth (cm)

** predicts volume; converted to mass with specific gravity of 0.40 (Alden, 1995).

† for *Tilia cordata*

†† for *Tilia americana*

‡ Correction factor 1.13